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Abstract

Atlantic lowland heath (ALH) is a priority conservation habitat in Western Europe, but restoration efforts have met with mixed success due to the complexity associated with replicating establishment conditions. By virtue of their impoverished, often acidic soils and geographic location in areas naturally occupied by ALH communities, former kaolinite mine sites may offer an excellent opportunity for heathland restoration. Using a chronosequence of former open cast kaolinite mines in SW England (0, 2, 27, and 150 years since mining ceased), we determined the ability of ALH vegetation and soils to re-establish naturally and in addition, for the three youngest sites, how the reinstatement of stored overburden affected heathland regeneration. Analysis of soil characteristics revealed major differences in the levels of acidity, organic content, and fertility between abandoned kaolinite sites and a nearby natural reference heath. Even 150 years after mining ceased, concentrations of all major soil nutrients and organic content were lower and pH higher, than undisturbed ALH. The reinstatement of overburden did little to improve soil quality, since all former kaolinite sites were dominated by mesotrophic grasses, rather than species characteristic of the target ALH community. We conclude that to maximise the potential of former open cast kaolinite sites for ALH re-establishment, changes in pre-and post-restoration management are required. These include modification of how overburden is stockpiled, while the addition of organic material, microbial communities, and sulphur (to reduce soil pH) to reinstated overburden are likely also essential interventions to facilitate successful ALH establishment.

Implications for Practice

- Successful restoration of former quarries requires a local propagule source, but such sources may be limited near large scale, open-cast mining operations.
- Reinstatement of stored overburden to former kaolinite sites does little to improve key soil characteristics and facilitate heathland restoration.
- Unlike most restorations using stored overburden, a requirement for low acidity poses a problem for heathland establishment and necessitates further manipulation of pH and soil nutrients to prevent dominance by mesotrophic grassland species.

Introduction

The Atlantic lowland heaths (ALH) of NW Europe are a distinct habitat characterised by a dominant heather (*Calluna* and *Erica* species), shrubby Fabaceae (i.e. *Ulex* and other Genisteae species) and distinctive graminoid (e.g. *Molinia caerulea*) community growing on low nutrient, acid soils (Gimingham 1972; Loidi et al. 2010). Most heathlands are the result of anthropogenic management imposed by periodic fire, grazing, or other disturbances, and as such, these habitats have an important cultural, as well as biodiversity and ecosystem service value (Mitchell et al. 2008; Pywell et al. 2011; Fagúndez 2012). In the UK alone, 133 conservation priority (UK Post-2010 Biodiversity Framework) plant and animal species are associated with ALH, including 47 invertebrates with either a restricted, or very restricted UK distribution, which are also internationally rare or endangered (Webb et al. 2009). Globally, heathlands face a number of threats, and due to a combination of changing management, atmospheric N deposition, and habitat loss associated with building development, the ALH habitats of western Europe, are particularly endangered (Fagúndez 2012; Bähring et al. 2017). For example, only one sixth of the lowland heath present in England from the early 19th century remains, much of this lost to urbanisation and agricultural intensification (Perrow & Davy, 2002; Webb et al. 2009). This dramatic decline has prompted the European Union and individual member states to adopt and implement various protection and restoration strategies for the habitat (JNCC, 2004; Pywell et al. 2011).

ALH requires at least four elements to establish and persist. These include: (i) low soil pH (2.8-3.9, Clarke 1997), (ii) low soil nutrient content (exchangeable calcium 80-159 µg Ca g^{-1}, exchangeable phosphorus < 10 µg P g^{-1} soil, Clarke 1997), (iii) propagule supply of dominant dwarf ericoid plants, and (iv) management to prevent succession to woodland (Kleijn et al. 2008; Martinez-Ruiz et al. 2007; Newton et al. 2009). On this basis, many different restoration techniques have been tested (Pywell et al. 1994; Clemente et al. 2016) but these can be categorised into two classes: those involving soil amelioration (e.g. nutrient addition, overturning), and those involving the selective addition of plants or seeds (Allison & Ausden, 2004; Walker et al. 2004a; Pywell et al. 2007; Glen et al. 2017). The results of many previous studies indicate the most important factor for successful heathland restoration is prior land use, as the most successful restorations are situated on former heathland (Walker et al. 2004b). Here, the removal of invasive scrub, coupled with the distribution of heather brash and native seeds (of local morphotype/genotype) cut and/or collected from local established adjacent heathland, have proved the most effective strategy (Walker et al. 2004b, 2007; Diaz et al. 2008). This approach not only provides the recipient community with necessary propagules and microbial symbionts, it also underscores the importance of soil biogeochemistry and the major contribution that the soil seed bank makes to heathland regeneration (Clarke 1993; Pywell et al. 1996; Fagúndez, 2013; Nussbaumer et al. 2016). This is not, however, a sustainable or practical technique to cover large areas or where there has been significant land use change, such as quarrying operations over many decades.

In many cases, effective restoration of former open cast quarries may be achieved by 'passive restoration' (Prach et al. 2001, 2013; Tropek et al. 2010). This approach has many advantages, including relatively rapid colonisation by local ecotypes of well-adapted species, with minimal economic costs to the mine operator (Prach & Hobbs 2008). Passive restoration seems to work best however, when the disturbed site is small, and surrounded by natural vegetation unaffected by the initial disturbance (Holl & Aide 2011; Prach et al. 2014).

Many OCM (Open Cast Mining) sites in the UK are located in areas naturally-dominated by heathland habitats, where high concentrations of mineral deposits, such as cassiterite, ilmenite, and kaolinite occur beneath the low nutrient, highly acidic, soils. This pattern is generally true of European ALH, where the habitat is most commonly associated with soils originating over low nutrient, sand and gravel beds (e.g. Belgium and the Netherlands), or igneous, typically granitic, intrusions (Scandinavia, Western France & UK). In SW England, ALH frequently coincides with deposits of the aluminosilicate mineral, kaolinite, a product of in-situ alteration of the plagioclase feldspar component of the granite intrusions that surface throughout Cornwall and west Devon. As a result, there has been a 300 year history of OCM kaolinite extraction in the region, supporting over 5% of the global extent of ALH habitat (Devon BAP, 2009).

In general, the nature and success of any restoration depend on the planning conditions imposed and the suitability of post-OCM conditions for plant establishment (Cooke & Johnson, 2002; Kuter, 2013). By its nature, OCM necessitates the removal of plant communities and underlying material to expose commercially-extractable minerals. The topsoil and underlying sediments (overburden) are mixed and stockpiled, destroying natural soil structure. Often, there is not enough original topsoil to cover the area left after extraction (Merino-Martín et al. 2017). Most problematic however, is the long period (between 1-20 years) of overburden storage, since over time, sub-surface soil layers in storage berms develop sub- or anoxic conditions, causing changes in microbial communities and further deterioration of soil structure and quality (Golos et al. 2016; Merino-Martín et al. 2017). In the specific case of ALH restoration, the combination of stockpiling methods and age since removal means that the availability of donor soils containing seed of the community dominant, *Calluna vulgaris*, is often limited (Pywell et al. 2002). Even if seeds are present, the altered characteristics and microbial communities of donor soils degraded by storage can reduce subsequent *Calluna* seedling establishment (Bossuyt & Honnay 2008). *C. vulgaris* can take 25-55 years to colonize mine spoil after cessation of mining operations (Roberts et al. 1982). In addition, the seeds of other component plant species, including many rare heathland specialists, are poorly represented in the overburden seed bank compared to their contribution in the natural ALH community (Bakker & Berendse 1999). The decline of the seed bank and loss of soil structure and microbial community in stored overburden underscores why the most successful attempts to restore ALH have been on former heathland sites. In these cases, some vestige of pre-disturbance soil propagule availability, microbial community, and soil biogeochemistry remains (Pywell et al. 1996; Walker et al. 2004b, 2007, Diaz et al. 2008; Wubs et al. 2018).

The aim of this case-study was to determine whether, and how quickly, after kaolinite mining has ceased, plant communities approach those of an undisturbed, target, ALH community. We also investigate temporal changes in the establishment of plant communities to see how closely the vegetation of former kaolinite extraction sites followed observed changes in soil quality and how quickly a post-OCM site would converge with typical ALH. In doing so, we test the hypotheses that even without any active attempt to ameliorate overburden, given sufficient time, it is possible to re-instate ALH following kaolinite OCM.

Methods

Study Sites

Located on the periphery of the Dartmoor National Park, Devon, SW England, commercial OCM kaolin extraction has taken place at Headon China Clay Works (50.2510°N, 03.5930°W) since 1855. The quarry offers a sequential series of sites where kaolinite extraction ceased in 1868, 1990, 2013 and 2015. As described by De Palma *et al.* (2018), this provides a space-for-time substitution (where spatial comparisons are made to infer temporal change) under a Control-Impact model with the associated limitations. All locations were therefore at a similar altitude and experienced similar climate, although the 1990 and 2015 sites were north facing and the 2013 south facing slope. The 1868 site was also south facing, but had a steeper slope angle (± 66%) compared to the younger sites (± 30%). While the 1868 site received no known post-OCM interventions, the 1990, 2013 and 2015 sites were covered in overburden stockpiled outside in large mounds approximately 6 m deep for 5 years, in order to help stabilise slopes. The nearby Trendlebere Down Nature Reserve (50.3641°N, 03.4424°W) was selected as a typical ALH reference site, as it had no history of mining, but similar slope, aspect, altitude, and geomorphology to the commercial quarry prior to kaolinite extraction. At the time the soil and vegetation surveys were undertaken, these sites were 147, 27, 2 and 0 years old, respectively.

Soil sampling and analysis

In summer 2015, ten sampling points at each restoration site were determined using the 'W-walk' method (JNCC, 2004). In each, a starting point was selected randomly and a quadrat (0.5 m × 0.5 m) placed to determine the first sampling position. The sample points were 20 m apart, and the total distance walked was 180 m. A 30 cm soil core was taken from the left corner of each quadrat using a soil coring kit (Eijkelkamp Agrisearch Equipment BV). The O horizon (~15 cm) was sampled from the cores and subsequently dried in a desiccator set to 65°C, disaggregated, sieved (2 mm mesh) and stored before analysis.

For pH analysis, 10 g of soil in 50 ml deionised water was mixed for 15 minutes with a magnetic stirrer, left to settle and determined using a Hanna 991001 pH and temperature probe (Jones Jr, 2001). As a proxy for organic matter, loss on ignition was used to quantify soil carbon content, with (~5 g) samples dried at 105°C for 1 hour, weighed and ashed at 400°C for 2 hours in a Gallenkamp hotspot furnace (Jones Jr, 2001). Mineral elements were extracted using the Mehlich 3 method (Jones Jr, 2001), whereby an extraction solution (30 ml) was added to each soil sample (3 g) in centrifuge tubes and mixed on a reciprocating mechanical shaker at 200 rpm for 5 minutes. Samples were subsequently filtered through Whatman 42 filter papers, and the filtrate retained in the dark until analysis. The Na, K, Mg, Ca and P concentration of the extracted solution was analysed using a Thermo Scientific iCAP7400 ICP-OES instrument.

To assess soil nitrate/nitrite concentrations, 3 g samples were digested in 30 ml of 0.01 M calcium sulphate, shaken on a reciprocal shaker for 15 mins at 180 rpm, and filtered through a Whatman 42 filter paper, followed by a cadmium reduction reaction and quantification by colourimetry (HACH DR/890) (Jones Jr, 2001). Cation exchange capacity, a measure of soil ability to retain key nutrients in 'plant-available' form, was quantified using the sodium acetate method (Jones Jr, 2001). We applied One-Way ANOVA, with a Welch's correction for unequal variances, to explore how these key soil chemical parameters varied according to the factor 'time since restoration'.

Vegetation sampling and analysis

Within each of the ten 0.5m × 0.5m quadrats positioned along the sample transect, species presence and absence was quantified (0 – absent, present – 1), and an NMDS using the Raup-Crick distance used to visualise variation in community patterns between sites (Clarke, 1993; Zuur, Ieno & Smith, 2007). Analysis was performed in three dimensions using metaMDS and ordiellipse to highlight groupings in the 'vegan' (Oksanen, 2015) package in 'R' v.3.5.2. Once the communities were plotted onto an ordination plot, the physical characteristics of the soil were overlaid as vectors (for variables where $P \leq 0.001$). Lines pointed in the same direction are positively correlated to each other (Zuur, Ieno & Smith, 2007). This enabled interpretation of the significant physical factors and how they were aligned with the various communities. An ANOSIM was performed in the 'vegan' (Oksanen, 2015) package in 'R' v.3.5.2 to examine variation in plant community composition between restoration treatments.

Results

Soil chemistry

We found little evidence to support the hypothesis that even several decades after OCM terminated, soils on former kaolinite sites would transition naturally towards soils favouring an ALH community. Even at the oldest (147-year-old) site, key aspects of soil chemistry were very different from the Trendlebere Down heathland (Table 1). Concentrations of major elements (Na, Ca, K and Mg) were generally an order of magnitude lower at the 1868 site, and soil P and NO_3 concentrations were 28% and 20% respectively of those in established heathland. With the exception of NO_3, the restored sites had lower major element levels; less than 25% of the reference site. Established heathland soil was also more acidic (pH 3.8), and had considerably higher organic matter content (67.5% OM) than all former OCM sites (pH 4.5-4.9, <6% OM), showing that the addition of stockpiled soil to the 1990, 2013, and 2015 sites had minimal beneficial impact on soil chemistry (Table 1).

Changes in vegetation community composition

Multivariate analysis revealed considerable variation in the plant community characteristics between each site (global $R_{ANOSIM} = 0.496$, $P < 0.001$), (Fig 1). The reference ALH community at Trendlebere was tightly clustered around the major defining plant species for this habitat (i.e. *Calluna*, *Molinia*, and *Erica tetralix*), these species also being more abundant here than any other site. The former OCM sites were less tightly clustered around distinct species; the 1868 site in particular showed broad overlap across many different plants, most uncharacteristic of typical ALH communities (specifically, the graminoids *Deschampsia, Festuca*, and *Juncus*, and the forbs *Potentilla* and *Galium*; species more commonly associated with acid grasslands). Nonetheless, *Calluna* and *Molinia* at the 1868 site achieved the highest abundance recorded at any former kaolinite mine location.

The 1990 and 2013 sites were dominated by Poaceae species characteristic of acid and mesotrophic grasslands (e.g. *Deschampsia flexuosa* and *Festuca rubra*), although the position of the 1990 cluster in the nMDS reflects that the contribution of both *Calluna* and *Molinia* to the community was much greater here, than at the younger 2013 site. Also, of note is the fact that *Ulex europaeus* was considerably more abundant at the 1990 site than any other location (although the presence of this N-fixing legume appeared to have little impact on soil NO_3). The 1990 site had the tightest cluster of all the OCM restoration sites. The 2015 site clustered around *Agrostis capillaris* (Fig 1), reflecting that quadrats here were dominated by bare ground and had no ALH-characteristic plants present. The most important environmental factors dictating plant community composition was the addition of overburden; K and P, and time ($P \geq 0.001$) (Fig 1).

Discussion

Although we emphasise that this case-study lacks true replication, our results nevertheless corroborate the general view that effective ALH restoration is a long-term process with little or no guarantee of success (Miller et al. 2017). Indeed, even after nearly 150 years (albeit with minimal additional management; i.e. grazing by livestock), soil chemistry failed to approach the levels of acidity, organic content, CEC or key soil nutrients characteristic of, and important in, heathland soil (Clarke 1993; Green et al. 2015). Similarly, although some species typical of established ALH were abundant in the 1868 site, the community was also characterised by species representative of acid or mesotrophic grasslands. There seems little potential therefore, to expect long-term, natural ALH recovery on the many kaolinite open cast mines located in regions where this habitat is most common, and especially where restoration occurs alongside active mining. Instead, and like many OCM sites globally, heathland restoration can likely only be facilitated by further interventions after mining operations cease (Holmes. 2001: Benigno et al. 2013; Clemente et al. 2016; Glen et al. 2017).

One commonly-applied approach is to reinstate stockpiled overburden onto former OCM sites, but the results suggest this practice did little to facilitate any improvement in key soil characteristics, or subsequent establishment of plant species typical of the target ALH community. Even on the 25-year-old (1990) site, organic content and pH of the reinstated overburden had little in common with those in nearby natural ALH. In theory, the use of topsoil provides a source of native seed, mycorrhizal and bacterial symbionts with which to facilitate plant community restoration (Muñoz-Rojas et al. 2016; Wubs et al. 2018). In practice, however, suitable topsoils are scarce and overburden (topsoil mixed with underlying mineral horizons) is stockpiled into large mounds to reduce footprint on the mine site, a procedure that diminishes key properties over relatively short periods (Golos et al. 2016). For the most part however, recent studies reporting the impact of soil stockpiling on restoration have focussed on (generally negative) changes in the soil seed bank (Dickie et al. 1988; Rokich et al. 2000) or soil microbial community (Harris et al. 1989; Poncelet et al. 2014). It may be the case however, that soil nutrients are less impacted by storage (Abdul-Karrem & McRae. 1984; Strohmeyer 1999).

A deficiency in the major macronutrients (NPK) required for plant establishment and growth in stored overburden and kaolinite mine waste is nonetheless well known (Marrs et al. 1981; Coppin & Bradshaw 1982). Phosphorus and potassium concentrations in the sites were considerably lower (i.e. less than 20%), even 25 years after overburden had been reinstated, than in the adjacent target community. Soil nitrate was, however, substantially higher in sites with overburden (1990, 2013, 2015) than in the 1868 site where no interventions were undertaken after OCM ceased. Other important heathland macronutrients, including Mg, Na and Ca (Clarke 1993, 1997), were frequently present at concentrations less than one-tenth of those seen in the adjacent ALH site. Removal and mixing of thin heathland topsoils with the mineral soils that underlie them before mining inevitably dilutes soil nutrients; subsequent storage and leaching from a generally coarse-grained overburden, further diminishes fertility. Reinstatement of a nutrient limited, mineral overburden where the symbiotic soil microflora plants require to extract nutrients from low fertility heathland soils are now absent, unsurprisingly limits establishment of heathland specialists, even if propagules are available (Diaz et al. 2006). To compound the problem, the low water retention capacity of coarse-grained, low organic content mineral overburden increases substantially the risk of plant mortality and reduced growth during drought (Machado et al. 2013; Bateman et al. 2018). Although relatively uncommon in SW England, future climate scenarios predict increased frequency of warm, dry summers, including extreme heatwaves and drought (Guillod et al. 2018).

Unlike the majority of mine rehabilitation studies where the low pH associated with overburden poses a major problem for plant community restoration (Abdul-Kareem & McRae 1984; Malik & Scullion 1998), none of the former kaolinite sites studied were as acidic as natural ALH. Low soil pH is critical for the establishment of the ericoid shrubs that characterise lowland heaths (Pywell et al. 1994; Marrs et al. 1998). Moreover, low pH often results in loss of cations from soils; Green et al. (2015) for example, reported a positive correlation between pH and concentrations of extractable K, Ca & Mg, but a negative association with phosphate. In-turn, soil concentrations of many elements affects the bioavailability of other key nutrients and also influence greatly the growth of species that might otherwise outcompete the target heathland species. Green et al. (2015) describe how at higher pH (5 or above), the vegetation of restored heathland sites was dominated by *Agrostis* *capillaris* and note how control of this highly competitive species is key to *Calluna* and *Erica cinerea* establishment. Similarly, our results show how *Agrostis capillaris*, along with at least one other mesotrophic grass species, was dominant on the 1868, 1990 and 2013 sites where soil pH remained above 4.7.

The failure of key soil characteristics or plant community composition throughout our chronosequence to trend towards those associated with the adjacent natural ALH strongly suggest that even where stored overburden is used, further manipulation is required. Benefits may accrue from reduction in overburden storage times and the depth of stockpiles (reducing compaction), and regular addition of organic material to retain soil meso-fauna and microbial populations and function, and water holding capacity during storage (Dickie et al. 1988; Rokich et al. 2000; Ngugi et al. 2018). Following reinstatement, further addition of organic matter to overburden is desirable for the same reasons (Smith & Read 2010; Muñoz-Rojas et al. 2016) and the potential enhancement of nitrogen cycling rates (Van Vuuren et al. 1992), while fertilizer application can also encourage plant establishment and growth, and concomitant benefits to soil biota (Ngugi et al. 2018). More specific to heathland restoration, soil pH is effectively reduced by the application of sulphur, with the additional benefit of increasing the bioavailability of phosphate without the need for fertilizer application (Green et al. 2015). Heather establishment is also strongly dependent on symbiotic interactions with ericoid mycorrhizal fungi (ERM) that do not respond well to long-term soil storage (Smith & Read 2010). Consequently, the introduction of essential ERM to the soil may be essential to effective ALH restoration where former kaolinite OCMs are covered with overburden stored for long periods. Taken together therefore, we conclude that effective restoration of ALH communities on former kaolinite quarries requires multiple interventions that address the limiting effects of low soil fertility, relatively high soil pH, propagule limitation, and an absence of soil micro- and macro-biota. Time alone is insufficient to facilitate these changes.

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TABLE 1 Comparison of mean (±SE, n=10) soil chemical conditions at an undisturbed Atlantic lowland heathland site (Trendlebere Down, Devon, UK – TBD) and former opencast kaolinite mine sites located in an adjacent commercial mine. The date when mining ceased at each site is given, and for the 1990, 2013 & 2015 sites, the termination of operations was followed by the replacement of stored overburden. The results of a one-way ANOVA of soil param are given, with different letters in superscript indicating significant difference in variance (P<0.05) between site means, following Tukey's paired comparisons. CEC = Cation Exchange Capacity

Site		Na ($\mu g\ g^{-1}$)	Ca ($\mu g\ g^{-1}$)	K ($\mu g\ g^{-1}$)	Mg ($\mu g\ g^{-1}$)	P ($\mu g\ g^{-1}$)	NO_3 ($\mu g\ g^{-1}$)	pH	Organic Content (%)	CEC (*mEq /100g*)
TBD	**Mean**	**16.5^{A}**	**73.2^{A}**	**48.2^{A}**	**56.4^{A}**	**8.1^{A}**	**2.0^{A}**	**3.8^{A}**	**67.5^{A}**	**76.7^{A}**
	(SE)	1.2	5.8	4.3	4.7	0.9	0.4	0.0	5.8	3.7
1868	**Mean**	**1.9^{B}**	**7.3^{B}**	**6.5^{B}**	**4.3^{B}**	**2.0^{B}**	**0.4^{B}**	**4.9^{B}**	**3.4^{B}**	**8.5^{B}**
	(SE)	0.3	1.7	1.1	0.7	1.2	0.1	0.1	0.6	0.6
1990	**Mean**	**2.7^{B}**	**7.2^{B}**	**9.2^{B}**	**3.9^{B}**	**2.0^{B}**	**1.8^{A}**	**4.7^{BC}**	**5.9^{B}**	**17.6^{B}**
	(SE)	0.1	3.3	2.0	0.7	0.1	0.2	0.1	0.3	1.5
2013	**Mean**	**1.5^{B}**	**16.4^{B}**	**4.1^{B}**	**3.9^{B}**	**2.0^{B}**	**1.1^{AB}**	**4.9^{B}**	**3.8^{B}**	**10.1^{B}**
	(SE)	0.1	4.1	0.5	0.6	0.1	0.3	0.1	0.5	1.5
2015	**Mean**	**1.6^{B}**	**9.9^{B}**	**5.6^{B}**	**3.5^{B}**	**0.8^{B}**	**1.4^{AB}**	**4.5^{C}**	**3.9^{B}**	**11.4^{B}**
	(SE)	0.2	3.8	0.6	0.4	0.2	0.4	0.1	0.6	1.5
ANOVA	**$F_{(4,45)}$**	**120.1**	**32.16**	**65.5**	**103.5**	**21.36**	**4.85**	**38.19**	**168.6**	**118.5**
	P	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001

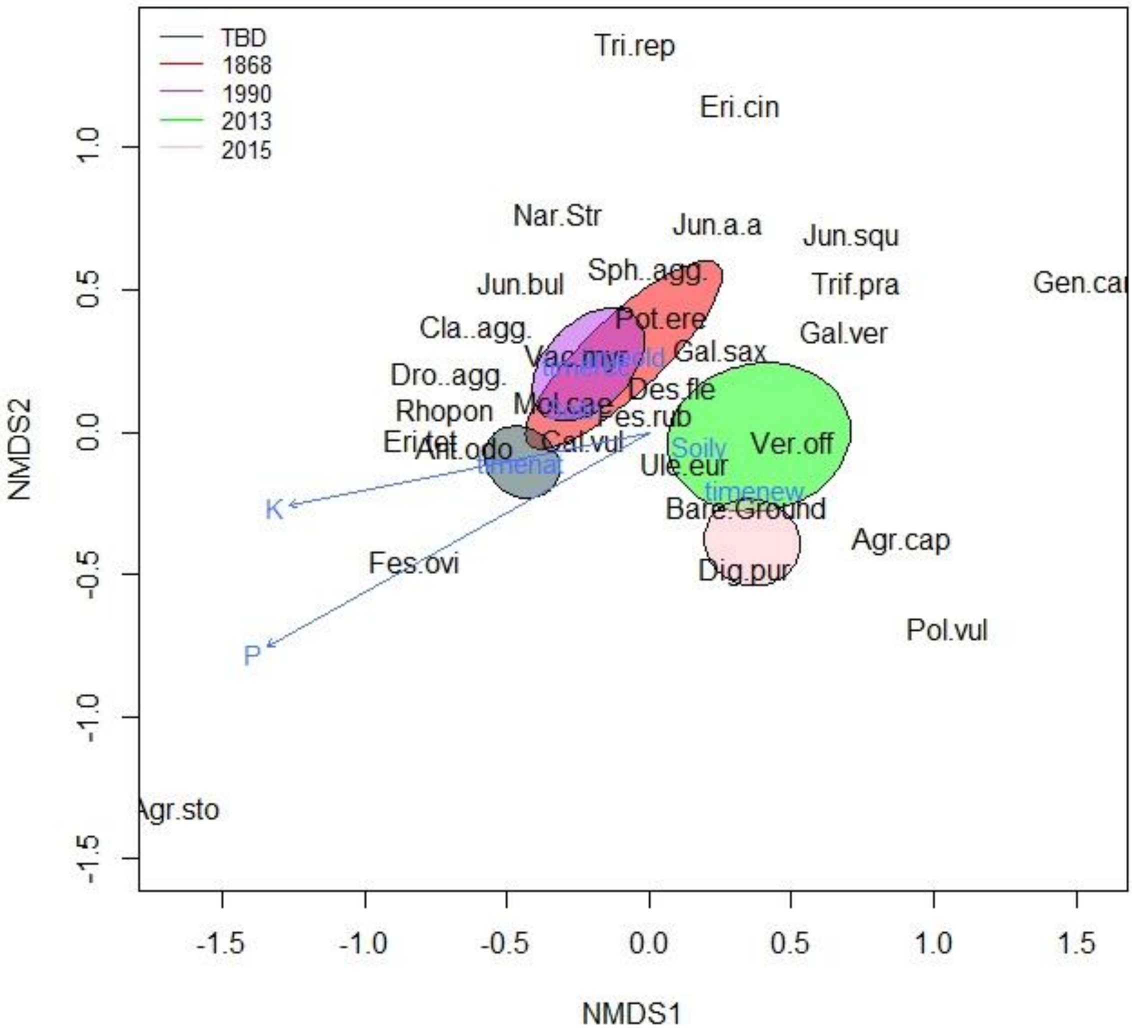

Figure 1: nMDS of the presence- absence data plant community of an undisturbed ALH site (TBD) and restored kaolinite mine sites (shown in 2 dimensions for ease of visualisation). The date labels denote the year when mining ceased. Stress =0.11. Ordiellipse are present to show the overlap of the communities. The vectors are significant environmental factors ($P \geq 0.001$)

Key to environmental factors: Timenatural, TBD; Timeold, 1868; Timerecent, 1990; timenew, 2013,2015; Soiln, overburden not added; Soily, overburden added.

Key to plant species: Agr cap, *Agrostis capillaris*: Agr sto, *Agrostis stolonifera*: Ant odo, *Anthoxanthum odoraturm*: Des fle, *Deschampsia flexuosa*: Fes ovi, *Festuca ovina*: Fes rub, *Festuca rubra*: Mol cae, *Molinia caerulea:* Nar str, *Nardus stricta*: Cal vul, *Calluna vulgaris*:

Eri tet, *Erica tetralix*: Eri cin, *Erica cinerea*: Rhopon, *Rhododendron ponticum* Vac myr, *Vaccinium myrtillus*: Ule eur, *Ulex europaeus*: Pot ere, *Potentilla erecta*: Ver off, *Veronica officinalis*: Gen cam, *Gentianella campestris*: Pol vul, *Polygala vulgaris*: Gal ver, *Galium verum*: Gal sax, *Galium saxatile*: Tri rep, *Trifolium repens*: Tri pra, *Trifolium pratense*: Dig pur, *Digitalis purpurea*: Dro agg, *Drosera* (agg): Jun a.a, *Juncus articulatus*: Jun bul, *Juncus bulbosus*: Jun squ, *Juncus squarrosus*: Sph agg, *Sphagnum* (agg): Cla agg, *Cladonia* (agg).